DOI:
10.1016/j.envint.2017.01.019

Environment International 101 (2017) 143–157

Contents lists available at ScienceDirect

Environment International

journal homepage: www.elsevier.com/locate/envint

Exposure to air pollutants during commuting in London: Are there inequalities among different socio-economic groups?

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ARTICLE INFO

Article history:
Received 10 October 2016
Received in revised form 4 January 2017
Accepted 27 January 2017
Available online 8 February 2017

Keywords:
Personal exposure
Environmental justice
Travel mode
Particulate matter
Ultrafine particles
Black carbon

ABSTRACT

People with low income often experience higher exposures to air pollutants. We compared the exposure to particulate matter (PM_1, $PM_{2.5}$ and PM_{10}), Black Carbon (BC) and ultrafine particles (PNCs; 0.02–1 μm) for typical commutes by car, bus and underground from 4 London areas with different levels of income deprivation (G_1 to G_4, from most to least deprived). The highest BC and PM concentrations were found in G_1 while the highest PNC in G_3. Lowest concentrations for all pollutants were observed in G_2. We found no systematic relationship between income deprivation and pollutant concentrations, suggesting that differences between transport modes are a stronger influence. The underground showed the highest PM concentrations, followed by buses and a much lower concentrations in cars. BC concentrations in the underground were overestimated due to Fe interference. BC concentrations were also higher in buses than cars because of a lower infiltration of outside pollutants into the car cabin. PNCs were highest in buses, closely followed by cars, but lowest in underground due to the absence of combustion sources. Concentration in the road modes (car and bus) were governed by the traffic conditions (such as traffic flow interruptions) at the specific road section. Exposures were reduced in trains with non-openable windows compared to those with openable windows. People from less income-deprived areas have a predominant use of car, receiving the lowest doses (RDD < 1 μg h^{-1}) during commute but generating the largest emissions per commuter. Conversely, commuters from high income-deprived areas have a major reliance on the bus, receiving higher exposures (RDD between 1.52 and 3.49 μg h^{-1}) while generating less emission per person. These findings suggest an aspect of environmental injustice and a need to incorporate the socioeconomic dimension in life-course exposure assessments.

1. Introduction

Air pollution is considered a major threat to human health because of its link to an increased mortality and loss of disability-adjusted life years (GBD 2013 Risk Factor Collaborators, 2015). Combustion emissions, especially particles in various size ranges, are suspected to be particularly harmful (Heal et al., 2012; HEI Panel on the Health Effects of Traffic-Related Air Pollution, 2010; WHO, 2013). Black carbon (BC) is considered a better tracer of traffic emissions than particulate matter (PM) mass (Reche et al., 2011; WHO, 2012), especially for diesel-fuelled vehicles. Owing to their size, ultrafine particles (<100 nm) may affect human health more strongly than larger-sized particles (Chen et al., 2016a, 2016b; Kumar et al., 2014; Lanzinger et al., 2016) and should be included in exposure assessments next to other pollutants.

Commuters are particularly affected by traffic-related air pollutants owing to their proximity to the source. BC and particle number concentrations (PNCs) represent ultrafine particles, which decrease exponentially downwind away from the road/highway (Fujitani et al., 2012; Kim et al., 2002; Zhu et al., 2006; Zhu et al., 2002). Such gradients are much weaker for PM_{10} ($PM \leq 10$ μm) and $PM_{2.5}$ ($PM \leq 2.5$ μm; Goel and Kumar, 2016; Kumar and Goel, 2016). Stationary monitoring stations provide a general view of actual fluctuation in air pollutants to which inhabitants are exposed (Chen et al., 2016a, 2016b; Reche et al., 2011). Such monitoring networks only provide a partial insight in personal exposure since this differs greatly with activity, location and time spent on each activity (Bekö et al., 2015; Buonanno et al., 2013; Rivas et al., 2016). Therefore, exposure assessment during commuting deserves special attention.

The miniaturisation of air pollution monitors has allowed the proliferation of personal measurements studies in different transport microenvironments over the few last years (Table S1). The studies have shown that commuters come in contact with highly variable concentrations of atmospheric pollutants and face short-time extreme peak concentrations that results in significant contributions by commuting to

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http://dx.doi.org/10.1016/j.envint.2017.01.019
0160-4120/

the total daily exposure (12–32% of daily exposure; Dons et al., 2011; Rivas et al., 2016; Williams and Knibbs, 2016). Further, the exposure during commuting is highly affected by individual mode of transport. Comparison among studies is challenging owing to variability in the methods used for sampling and different conditions in each transport mode (such as ventilation rates and fuel type; Goel and Kumar, 2015a; Karanasiou et al., 2014; Kaur et al., 2007). Moreno et al. (2015a, 2015b) reported the following hierarchy for PNC in different transport microenvironments with data from various studies: urban background < underground < tram < walking in a suburban main road < walking and cycling in the city centre < bus. However, this hierarchy might differ for other pollutants. For example, the highest PM concentrations are expected to be found in the underground (Adams et al., 2001; Martins et al., 2016a). Concentrations of $PM_{2.5}$ were lower in buses than in cars in Barcelona (de Nazelle et al., 2012) and Arnhem (Zuurbier et al., 2010), but a reverse situation was reported in London (Adams et al., 2001) and Dublin (McNabola et al., 2008). Consequently, more studies, such as this work, are needed to identify the parameters affecting pollutant concentrations in different transport microenvironments.

The distribution of air pollutants has been found to be inequitable, with people living in most deprived areas generally suffering from higher concentrations of air pollutants (Fecht et al., 2015; Kingham et al., 2007; Wheeler and Ben-Shlomo, 2005; WHO, 2010; Yu and Stuart, 2016). The field of environmental justice has been notably explored in the U.S.A., where poorer people or ethnic minorities are exposed to higher air pollutant concentrations (Bullard, 2015; Hackbarth et al., 2011; Houston et al., 2004; Yu and Stuart, 2016). A smaller number of studies are available for European countries (Barceló et al., 2009; Moreno-Jiménez et al., 2016), sometimes with inconclusive results (Padilla et al., 2014; Wheeler and Ben-Shlomo, 2005) or reverse between socioeconomic status and air pollutants concentrations (Forastiere et al., 2007; Germani et al., 2014). In the UK, Fecht et al. (2015) found an association between PM_{10} concentrations and deprivation in England, with the most vulnerable groups encountering higher concentrations. However, in a between neighbourhood comparison, both Fecht et al. (2015) and Goodman et al. (2011) observed a nonlinear relationship as people in the higher social class would accept high levels of air pollution to take advantage of the benefits offered in city central areas. Jephcote and Chen (2012) found that children in lower social class households in Leicester tend to live in areas experiencing high levels of road transport emissions which were caused to a substantial extent by the private transport of affluent communities living in areas with low emissions.

Unlike available studies, this work assesses the inequalities in exposure to air pollutants during commuting using real-time personal measurements, thus providing a precise input of exposure concentrations. The main objective of this work is to determine if there are inequalities related to income deprivation in the exposure during commuting to different fractions of PM, BC and PNC in London. To this end, different routes in different transport modes were assessed, with the routes being typical commuting routes for inhabitants from 4 areas with different level of income deprivation (G_1 to G_4, representing from most to least deprived). Furthermore, we have assessed the differences between transport modes (car, underground and bus) and different daytime periods (morning and afternoon rush hour, midday non-rush hour) in order to identify the main drivers of exposure during commuting.

2. Methodology

2.1. Study area

The study was carried out in Greater London (Fig. S1), which has an area of 1572 km^2 and around 8 million inhabitants (Office for National Statistics, 2014), making it one of the largest cities in Europe. In March 2016, London counted 3.3 million registered vehicles (2098 veh km^{-2}), of which 2.8 million were cars (1809 cars km^{-2}; Department for Transport, 2016).

2.2. Route selection

2.2.1. Datasets used for the route selection

Two different datasets were used for the selection of the origin and destination of our routes. One was the 2015 Index of Multiple Deprivation (IMD; Department for Communities and Local Government, 2015), which is the official measure of relative deprivation for small areas in England. The index consists of a basket of indicators from seven domains (which measure different dimensions of deprivation) to produce an overall relative measure of deprivation. The second dataset was the 2011 Census Special Workplace Statistics (Census Support Flow Data, 2011), which includes commuting counts (location of usual residence and place of work by method of travel to work). Both datasets report statistics at a small area level, the Lower Layer Super Output Area (LSOA), which represent homogeneous neighbourhoods in terms of key demographic and socioeconomic characteristics.

From the seven domains of IMD (Smith et al., 2015), we selected the Income Deprivation Score to classify the LSOA areas within the Greater London into 4 different groups (G_1 to G_4, from most to least deprived, with G_1 and G_4 representing the 10% most and least deprived, respectively; Table S2). The Income Deprivation domain is not an individual measure of affluence but identifies aspects of income deprivation at the small area level and was selected to ease replication in other countries. The spatial distribution of both the Income Deprivation and the IMD score are presented in Supplementary Information Figs. S2 and S3, respectively. There is a strong correlation across London for both indexes, suggesting that similar results could be expected if IMD would be used instead.

2.2.2. Selection of the origins, the destination and the routes

We aimed to select typical commutes for areas of residence with different levels of income deprivation. We selected one workplace area that is a frequent destination for commutes from origins in all deprivation classes (area with highest employment density). The destination point was within the City of London (LSOA name: City of London 001F), which is the financial district (Fig. 1).

For the single destination, we selected four origins, one in each deprivation class (Fig. 1). For each income group (G_1–G_4), we calculated the average Euclidean distance that the inhabitants commute in order to get to the selected destination, according to the origin-destination information reported in the Census Support Flow Data (2011). An increasing distance was observed from the most to the least deprived (Table 1). Afterward, a random LSOA at the corresponding average distance from the destination was selected as the origin for each of the four income categories. The origin point of the route was then chosen within the selected LSOAs, obtaining 4 origin-destination (O-D) pairs.

According to the Census Support Flow Data (2011), across all groups, the dominant modes were car (private), underground and bus (Table 1) and, accordingly, these three transport modes were assessed in this work. For each of the 4 O-D pairs, we monitored the fastest route for each transport mode (Fig. 1). Table S2 indicates the specifications for each of the routes (main roads used for car, and bus and underground lines). The same underground lines in opposite directions were taken for G_1 and G_3 (Northern line, with part of G_1 also in Victoria line) and for G_2 and G_4 (District line).

2.3. Instrumentation and sampling design

This work has been focused on the assessment of the exposure to particulate pollutants. Gaseous pollutants are also an important threat to human health, but for practical reasons and because of their potential health effect we selected to monitor PM_1, $PM_{2.5}$, PM_{10}, BC and PNC. A GRIMM EDM 107 (GRIMM Technologies Inc.) aerosol spectrometer

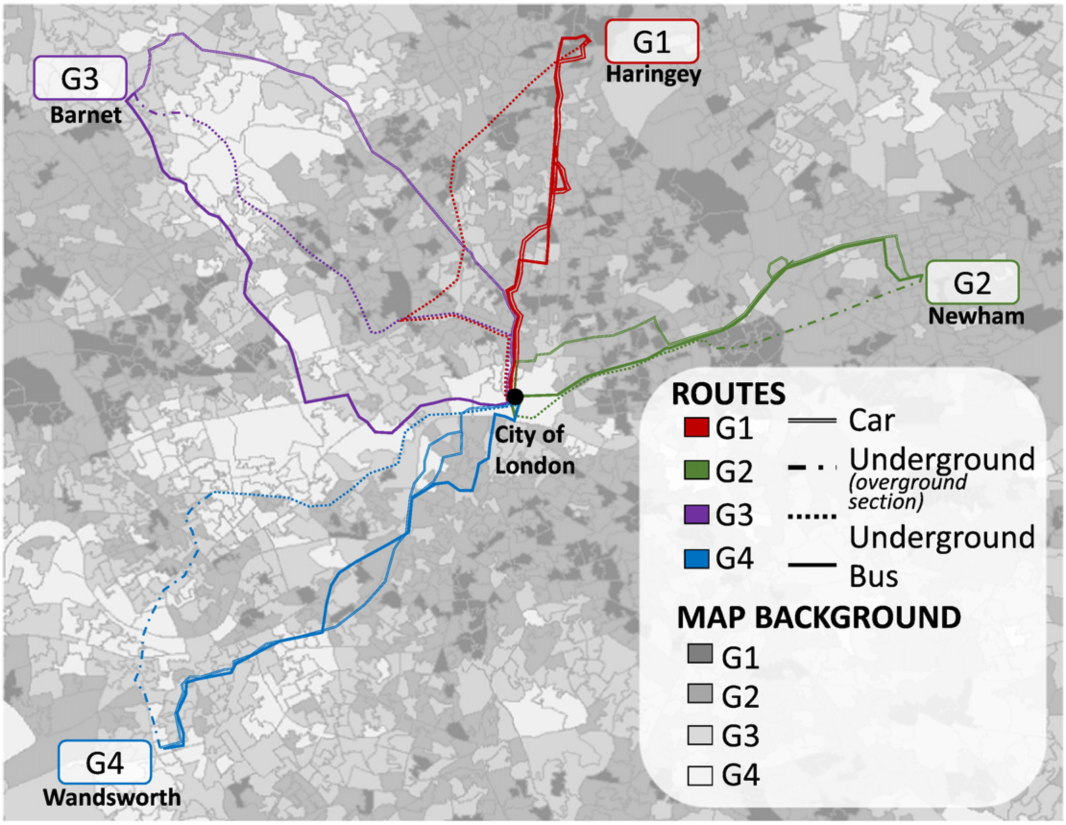

Fig. 1. Selected commuting routes within Greater London for measuring personal exposures to different air pollutants. The map shows the location of the destination and the four different origins as well as the routes for each mode of transport. The background indicates the income group to which the LSOA belongs.

was employed for personal PM measurements during commuting. The instrument provided PM mass concentrations in 31 different size channels at a time resolution of 6 s, but was afterwards averaged into 10 s to match the averaging period set on the rest of the instruments. Moreover, the GRIMM collects particles on a PTFE filter, allowing chemical and morphological analysis of the particles. This instrument has been widely used for monitoring PM concentration, including mobile measurements (Azarmi and Kumar, 2016; Grimm and Eatough, 2009).

PNC in the 0.02–1 μm size range were measured by the portable P-Trak 8525 (TSI Inc.), which has been successfully used for personal exposure monitoring (Aarnio et al., 2005; Delfino et al., 2005; Kaur et al., 2005). Traffic emissions contribute importantly to particles in the nucleation mode (<20 nm; Gidhagen et al., 2005; Kumar et al., 2008a, 2008b; Wehner et al., 2002), which are below the lower threshold for the particle size of the P-Trak. The deposition losses in our setting (length of the tube: 1.2 m, $T = 286.1$ K) for particles in the 20–100 nm range were always below 5%, and for particles between 100 and 1000 nm below 1%, according to the Hinds turbulent model (Hinds, 1999; Kumar et al., 2008b). Therefore, deposition losses in sampling tubes in our work have a small impact on final PNC measured by P-Trak. For this study, we set the averaging period for PNC data to 10s.

BC concentrations were monitored with a portable MicroAeth AE51 (AethLabs), which has been widely used for personal exposure assessment (Buonanno et al., 2013; de Nazelle et al., 2012; Dons et al., 2011; Moreno et al., 2015b; Rivas et al., 2016) and positively evaluated against reference stationary instruments (Viana et al., 2015). The effect of filter loading was minimised by replacing the filter strips before every trip and setting a flow rate of 100 ml min^{-1}. The time-base was set to 10 s. For high time resolution, instrumental optical and electronic noise can cause the attenuation (ATN) values to remain unchanged or slightly diminish from one timestamp to the next (the latter, leading to negative BC concentrations). Therefore, the original data was then post-processed with the Optimised Noise-reduction Averaging algorithm (ONA; Hagler et al., 2011), which smoothens the BC time series through a user-specified minimum change in attenuation (ΔATN_{min}).

The position of the field worker was continuously recorded on a second basis using a Global Positioning System (GPS; Garmin Oregon 350). The instruments were carried in a specially conditioned bag, with the inlets positioned at the breathing height of the field worker (Fig. S4). The bag was carried on the back while walking or standing in the underground or buses. If the field worker could have a seat, the bag was placed on the lap in such a way that the inlets were still at breathing

Table 1
Origin and destination LSOA, average Euclidean distance, income score for the selected origin LSOA and dominant transport modes for the different income groups.

ID	Origin LSOA name	Destination LSOA name	Avg. distance (km)	Origin income score[a]	Dominant mode (%)		
					1st	2nd	3rd
G_1	Haringey 012D	City of London 001F	7.7	0.353	Private (26.9)	Bus (25.0)	Underground (22.4)
G_2	Newham 011A		9.4	0.211	Private (29.8)	Underground (24.6)	Bus (18.7)
G_3	Barnet 037C		11.5	0.085	Private (36.1)	Underground (24.5)	Bus (11.7)
G_4	Wandsworth 025E		12.2	0.027	Private (37.4)	Underground (23.2)	Bus (6.9)

[a] Income score range for London: 0.007–0.388 (the highest the score, the most deprived).

height. During car measurements, the bag was securely placed on the front passenger seat.

The trips were generally bi-modal since short distances needed to be walked to complete the commuting (i.e. from origin/destination address to the nearest bus/underground station) and, therefore, included different sub-microenvironments. Besides commuting, 15 min of stationary measurements were carried out at the origin point before starting the trip, 15 min at the destination point, and 15 more minutes at the origin when back from the returning trip. This procedure allowed us to take into account the inter-day variability of background concentrations. We completed a time-location diary for every trip, where we registered the time for all the movements (e.g., start/end time of the trip, time when arriving at the bus stop/underground station) and identified the main locations/sub-microenvironments affecting the exposure during commuting.

All the car measurements were made with a petrol-fuelled Peugeot 208 Active 1.2 VTI 82 HP (registered in 2013). Car windows were kept closed and the ventilation settings were at 50% fan velocity and no air recirculation. We had no control over the ventilation conditions in the public transport modes (i.e. bus and underground) but we registered this information. For bus, we noted if the windows of the vehicle were open. The field worker always sat in the lower deck of the bus, on the first or second row of seats behind the rear door. If no seats were available, the field worker was standing within the same area. We considered two important variables for the underground measurements: (i) if windows were open or if they were non-openable and with mechanical ventilation; and (ii) whether the platform or train were located in or passing through an uncovered (open and above ground) or covered (underground) area.

2.4. Data collection and processing

The fieldwork was performed over a total of 40 days between 25 February and 17 June 2016 at 3 different daytime periods (starting at 07:45, 12:00 and 16:30 h local time). We monitored a total of 117 round trips (origin – destination – origin; corresponding to 232 one-way trips; Table 2). The trips were equally distributed among the 4 different income deprivation areas (25% each group) and among the day periods (morning = 32%, afternoon = 34%, evening = 33%). More trips in car and underground were monitored since those are the prevalent modes (car = 38%, underground = 41%, bus = 21%). Two return trips were cancelled owing to disruption on the underground and bus service. We measured a total of 225 h of trip monitoring data, from which 98.7% of PM, 99.4% of BC and 91.9% of PNC data was available for analysis. In addition, a total of 66 h data were also monitored at the origin and destination points.

We determined the PM respiratory deposition doses (RDD) for different PM fractions according to the methodology presented in our earlier work (Azarmi and Kumar, 2016; Kumar and Goel, 2016). In brief, RDD is estimated according to Eq. (1), which is adapted from ICRP (1994):

$$RDD = \sum VR_{i,a} \times DF_{j,k} \times C_{j,k} \tag{1}$$

where $VR_{i,a}$ is the ventilation rate for the i^{th} individual (depends on age and sex) during activity a; $DF_{j,k}$ and $PM_{j,k}$ are the Deposition Fraction and PM mass concentration for each of the j^{th} PM fraction in the transport sub-microenvironment k, respectively. Hourly RDD were calculated for each sub-microenvironment, as well as the RDD per trip when multiplying the hourly RDD for the time spent in each sub-microenvironment ($t_{i,k}$). Since DFs are not directly proportional to the mass concentration, these were calculated with the corresponding mass median diameter (MMD) according to equations presented in Hinds (1999). Table S3 provides the values of the different parameters and further equations used for the calculations of RDD.

The data management and statistical analyses were performed with the R statistical software (v 3.0.2, R Core Team, 2016) and the packages *openair* (Carslaw and Ropkins, 2012) and *dunn.test* (Dinno, 2015). ArcGIS 10.1 (Esri Inc.) was employed for the calculation of distances, and for generating the concentration maps.

2.5. SEM and EDS analyses

Total PM mass was collected on 47 mm PTFE filters using GRIMM 1.107. A total of 4 samples were collected, one for each transport mode and a blank filter for reference. After carbon coating, particles were characterised using a JEOL SEM (model JSM-7100F, Japan) with a spatial resolution of 1.2 nm at 30 kV and 3.0 nm at 1 kV. The JEOL SEM was equipped with EDS, thus being able to obtain information on morphology and elemental composition of the particles. Samples were

Table 2
Descriptive statistics showing the data availability, geometric mean (GM), geometric standard deviation (GSD) of pollutant concentrations on each of the routes, mode of transport and period of the day. # trips = number of one-way trips. n = number of data points (10 s average); d.u. = dimensionless unit.

	Data available				GM (GSD)				
	# trips	n (BC)	n (PNC)	n (PM)	BC μg m^{-3} (d.u.)	PNC # cm^{-3} (d.u.)	PM_1 μg m^{-3} (d.u.)	$PM_{2.5}$ μg m^{-3} (d.u.)	PM_{10} μg m^{-3} (d.u.)
Income area									
G_1	58	11,371§	15,553	17,179	5.0 (1.8)^{§,A,B}	9335 (1.7)^{D,E}	14.7 (2.8)^{G}	19.5 (3.1)^{I}	33.9 (3.6)^{L}
G_2	60	12,993§	18,341	18,873	3.3 (2.0)§	6273 (1.9)	9.1 (2.3)^{H}	11.5 (2.4)^{J}	20.9 (3.1)^{M}
G_3	57	14,429§	19,754	20,913	5.9 (1.9)^{§,A,C}	8734 (1.8)^{D,F}	12.4 (2.9)^{G}	16.4 (3.4)^{I,K}	29.0 (4.4)^{L}
G_4	57	15,623§	20,791	23,057	5.2 (1.9)^{§,B,C}	8640 (1.9)^{E,F}	10.4 (2.2)^{H}	13.5 (2.3)^{J,K}	23.9 (3.1)^{M}
Kruskal-Wallis test, (significance at $p < 0.05$):					$p < 0.01$	$p < 0.01$	$p = 0.01$	$p < 0.01$	$p = 0.02$
Mode									
Car	90	30,043	27,380	29,346	4.4 (2.5)	8639 (2.0)^{N}	6.7 (2.1)	7.3 (2)	8.2 (2.2)
Underground	95	26,101	24,538	26,624	9.8 (4.9)$^{\Delta}$	6694 (1.6)	23.3 (2.7)	34.5 (2.9)	68.4 (3.0)
Bus	47	24,373	22,521	24,052	5.4 (2.3)	9355 (1.8)^{N}	9.9 (1.8)	13.9 (1.7)	37.9 (2.1)
Kruskal-Wallis test, (significance at $p < 0.05$):					$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$	$p < 0.01$
Ratio underground/car (dimensionless):					2.2	0.8	3.5	4.7	8.4
Ratio bus/car (dimensionless):					1.2	1.1	1.5	1.9	4.6
Period									
Morning	74	18,105	24,545	26,643	5.7 (2.3)§	8989 (1.9)	13.5 (2.4)	17.8 (2.6)	32.4 (3.3)
Afternoon	80	18,389	25,090	26,986	4.1 (2.3)^{§,O}	7530 (1.8)^{P}	10.0 (2.6)^{Q}	13.0 (2.8)^{R}	22.6 (3.6)^{S}
Evening	78	17,922	26,532	24,804	4.7 (2.6)^{§,O}	7980 (1.8)^{P}	11.0 (2.7)^{Q}	14.1 (2.9)^{R}	24.9 (3.7)^{S}
Kruskal-Wallis test, (significance at $p < 0.05$):					$p < 0.01$	$p < 0.01$	$p < 0.03$	$p = 0.05$	$p = 0.07$

NOTE: Pairs sharing the same letter in their superscript are not significantly different at the 0.05 level according to Dunn's Test (run for the GM of each single trip).
§ Underground measurements were excluded.
Δ Probably overestimated because of Fe interference in the light absorption of the MicroAeth measurements.

scanned with a high-energy beam (5–15 kV) of electrons in a raster pattern. Due to the carbon coating, we were not able to examine the elemental and organic carbon on the samples. The analyses were carried out at the MicroStructural Studies Unit of the University of Surrey (UK).

3. Results and discussions

3.1. Differences in concentration in the overall trip

3.1.1. Among income groups

Table 2 shows the geometric mean (GM) and geometric standard deviation (GSD) for all the pollutants. The route corresponding to G_1 showed the highest concentrations for PNC (9335 cm^{-3}) and all PM fractions (PM_1 = 14.7 μg m^{-3}, $PM_{2.5}$ = 19.5 μg m^{-3}, PM_{10} = 33.9 μg m^{-3}), with the G_2 route having the lowest (PNC = 6273 cm^{-3}, PM_1 = 9.1 μg m^{-3}, $PM_{2.5}$ = 11.5 μg m^{-3} and PM_{10} = 20.9 μg m^{-3}). Hereafter, when referring to the general term "PM" we are indicating that the statement is followed by the three fractions (PM_1, $PM_{2.5}$ and PM_{10}). PM and PNC concentrations vary across the groups with a statistically significant difference (Kruskal-Wallis test, $p < 0.05$). From the pairwise Dunn's test (significant at $p < 0.05$), G_2 emerged with significantly lower PNCs (Table 2) unlike the other groups (G_1, G_3, G_4) that were not significantly different from each other. Regarding PM concentrations, differences for the pairs G_1–G_3 and G_2–G_4 in all fractions were not statistically significant, which suggests that PM concentrations were governed by other factors than income deprivation. Comparison between income areas in each mode of transport (Table S4), shows that global PM concentrations were dominated by the very high concentrations in the underground. The difference between the pairs G_1–G_3 and G_2–G_4 corresponds to the two different underground lines (in opposite directions): the Northern line for G_1 (partly, also Victoria line) and G_3 and the District line for G_2 and G_4. Concentrations in the Northern and Victoria lines were >3-times higher than in the District line for PM (Table 3). This aspect is further discussed in Section 3.2.

For BC concentrations, we excluded the underground measurements in the calculation of the global GM for each income group (Table 2). Measured BC concentrations are affected by iron (Fe) interference in the measurements (Moreno et al., 2015b) due to its absorbance of visible light at similar wavelengths (Gilardoni et al., 2011). In most environments, the light absorption from Fe is negligible compared to that of BC (Ballach et al., 2001; Heintzenberg, 1982) but the underground has a considerable presence of Fe (Fig. S5; Kam et al., 2013) causing an important overestimation of BC. Fe influence is demonstrated by the correlation between BC and all PM fractions, with an $r > 0.85$ (Pearson) in the underground measurements, but only $r \leq 0.40$ for the car and bus measurements (Table S5).

If looking separately by transport mode, we can observe a similar trend among the groups for all the pollutants and transport modes (Fig. 2, see Fig. S6 for PM_1). G_1 arises as the one showing the highest concentrations of all pollutants in the car and the underground modes. The bus route corresponding to G_3 was the most polluted (except for PM_{10}, with the highest concentrations observed in G_4). G_2 arise again as the routes with the lowest concentrations, with the only exception of PM in the car (G_3 had the lowest concentrations). However, although this was the hierarchy, the Dunn's test showed in many cases that the differences among some groups are not statistically significant (Table S4). The similarities in pollutant concentration between the pairs G_1–G_3 and G_2–G_4 in the underground, attributed to the two different lines monitored, can be clearly observed in Fig. 2.

To the knowledge of the authors, this is the first study to assess the difference in concentrations to what commuters are exposed to according to their socioeconomic status. Despite G_1 showing the highest concentrations of all the studied pollutants (except BC), there was no consistent decreasing or increasing trend between the concentrations and the income groups (Table 2). These observations indicate that concentrations are not solely determined by the income group but by other factors (such as the line in the case of the underground measurements).

Table S6 shows the concentrations measured *stationary* at the *origin* and *destination* points by income group, transport mode and period. Concentration at the origin of all pollutants varies across income groups with a statistically significant difference (Kruskal-Wallis test, $p < 0.05$) which points to a difference in the background concentrations. The origin point in G_1 showed the highest levels of air pollutants but this can be explained by heavy traffic on this particular street compared to the other origin points. PM concentrations at the destination were not statistically different between the groups (Table S6), indicating similar background conditions for the different income groups. However, we observed significant variation across the groups for BC and PNC concentrations. This might be due to different traffic intensities or the effect of meteorology on the different sampling days that might affect differently to each pollutant. For example, BC and PNCs are considered better traffic tracers (and therefore more directly affected by traffic emissions) than $PM_{2.5}$ or PM_{10} (although they are obviously influenced by traffic emissions; Reche et al., 2011; WHO, 2012).

3.1.2. Among transport modes

PM concentrations were significantly highest in the underground ($PM_{2.5}$ = 34.5 μg m^{-3}) for all fractions, followed by the bus ($PM_{2.5}$ = 13.9 μg m^{-3}) and being lowest in the car ($PM_{2.5}$ = 7.3 μg m^{-3}; Table 2). Taking car as the reference, PM_1, $PM_{2.5}$ and PM_{10} concentrations were up to 3.5-, 4.7- and 8.4-times higher in the underground and 1.5-, 1.9- and 4.6-times higher in the bus, respectively. Past studies also report much higher PM concentrations in underground trains and platforms compared with the other transport modes or surface platforms (Adams et al., 2001; Cartenì et al., 2015; Martins et al., 2016a; Seaton et al., 2005). Our physicochemical characterization suggested Fe as the main component in underground PM (>50% by weight). This is in line with previous studies suggesting its origin from mechanical abrasion between rails, wheels and brakes (Jung et al., 2012; Kam et al., 2013; Martins et al., 2016a, 2016b; Querol et al., 2012). Fe particles were characterised by a flake-shape morphology (Fig. S5) with sharp and irregular borders, as also described by Moreno et al. (2015a) in the underground of Barcelona. Si, Cu, Ca, Zn and K, identified by EDS in our underground sample, are also commonly found in underground systems (Aarnio et al., 2005; Jung et al., 2012; Loxham et al., 2013; Martins et al., 2016b; Nieuwenhuijsen et al., 2007). The mass median diameter (MMD) inside the underground trains is 3.5 μm, which is within the coarse fraction ($PM_{2.5-10}$). Despite the high concentrations, Seaton et al. (2005) concluded that concentrations in the underground were unlikely to represent a significant risk due to the different nature of the underground (mainly iron oxides) from the surface particles,

Table 3
GM and GSD of the pollutant concentrations inside the underground trains according to the line (d.u. = dimensionless unit). BC is overestimated in the underground due to Fe-interference.

Underground line	GM (GSD)				
	BC μg m^{-3} (d.u.)	PNC # cm^{-3} (d.u.)	PM_1 μg m^{-3} (d.u.)	$PM_{2.5}$ μg m^{-3} (d.u.)	PM_{10} μg m^{-3} (d.u.)
District	8.7 (3.2)	5448.4 (1.5)	17.5 (2.1)	23.8 (2.2)	46.0 (2.6)
Northern	40.3 (2.2)	7815.1 (1.5)	56.8 (1.9)	104.0 (2.0)	204.2 (1.8)
Victoria	62.6 (1.4)	6823.6 (1.3)	87.4 (1.4)	131.6 (1.4)	208.1 (1.4)

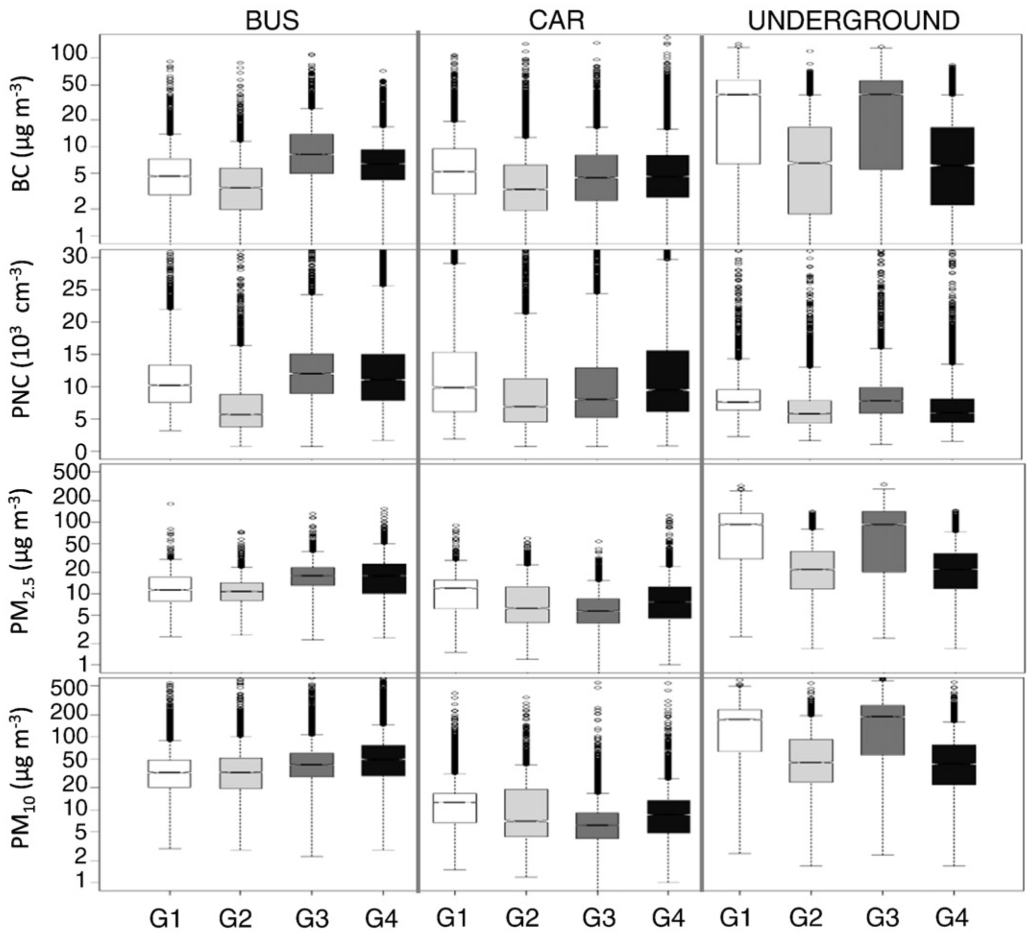

Fig. 2. Boxplot of BC, PNC, $PM_{2.5}$ and PM_{10} concentration measured at the different transport modes by income group. Boxes represent 25th and 75th percentile, central line the median, bars outside the box represent the 1.5× interquartile range, and circles are outliers. The notch displays the confidence interval around the median.

which are mostly combustion generated and suspected to pose a major risk (Kelly and Fussell, 2012).

Commuters travelling by bus were exposed to higher BC concentrations (BC = 5.4 μg m^{-3}) than those travelling by car with the windows closed (BC = 4.4 μg m^{-3}). This is due to a higher infiltration of outside pollutants into the bus because of open windows and frequent door opening. The fact that the air gets inside the car through the filters of the ventilation system hindered the entrance of particles, especially the coarse PM fraction ($PM_{2.5-10}$; Kumar and Goel, 2016). In fact, the PM_1 fraction corresponds to the 79.0% of the PM_{10} inside the car (Fig. 3). On the other hand, during the bus trips, PM_1 contributed only a 21.5% to PM_{10} inside the bus and therefore the bus trips are characterised by a very high proportion of coarse particles. The MMD inside the car was 2.80 μm, while for the bus was 11.4 μm. In car and bus modes, particles show a variety of morphologies such as agglomerates from vehicle exhaust, mineral components with crystal form, as well as the presence of sporadic fibres that may come from clothes worn by commuters or from the seat fabric. Fe was also the dominant element (Fig. S5), which is usually found in dust profiles of urban environments (Amato et al., 2009). The presence of Ba and Cu also indicate the contribution of non-exhaust emissions from the brake wear (Gietl et al., 2010; Thorpe and Harrison, 2008).

Regarding PNC, and opposed to PM, the underground had the lowest concentrations (6694 cm^{-3}) which have also been observed in other travel mode comparisons (Moreno et al., 2015b). This is due to the absence of combustion sources (such as vehicular emissions) in this environment. Although higher PNC were observed in the bus (9355 cm^{-3}), no statistically significant differences were observed with the car (8639 cm^{-3}; Table 2). Therefore, ultrafine particles might infiltrate more easily through the car filtering system, resulting in similar in-vehicle PNC concentration for bus and car modes.

Fig. 4 gives an example of a time-series of bus, car and underground measurements. The lowest BC and PM concentrations were observed in the car (grey background indicate in-vehicle measurements) and the presence of intermittent peaks might be associated with the traffic intensity and velocity. There is an obvious reduction of the concentration of coarse particles inside the car. However, coarse particles were very important in the bus measurements, with a very high short-time variability that can be attributed to the frequent opening of the doors and to the resuspension generated by the passengers. PM concentrations are perceptibly higher in the underground measurements. They reach to extreme peaks when the train goes underground, indicating a strong effect of the tunnels on PM concentrations (Adams et al., 2001; Seaton et al., 2005). This effect was also observed on PNC concentrations, however, the increase was less pronounced.

Globally in Greater London, a similar percentage of trips among the income groups were done by underground (Table 1). However, differences in the car and bus share were obvious. An increasing use of the car was observed from most (26.9%) to least (37.4%) deprived, being the opposite for the bus share (25.0% and 6.9%, most and least deprived, respectively; Table 1). The mode share may have implications on the exposures among commuters from areas with different income deprivation. Commuters from less deprived areas have a predominant use of the private car mode and therefore were exposed to the lowest concentrations. Contrary to that, a high proportion of people from the low income areas are using the bus which leads to higher exposures because of the higher concentrations (Table 2) as well as the longest trip duration (Table S7). Furthermore, private cars generate the highest

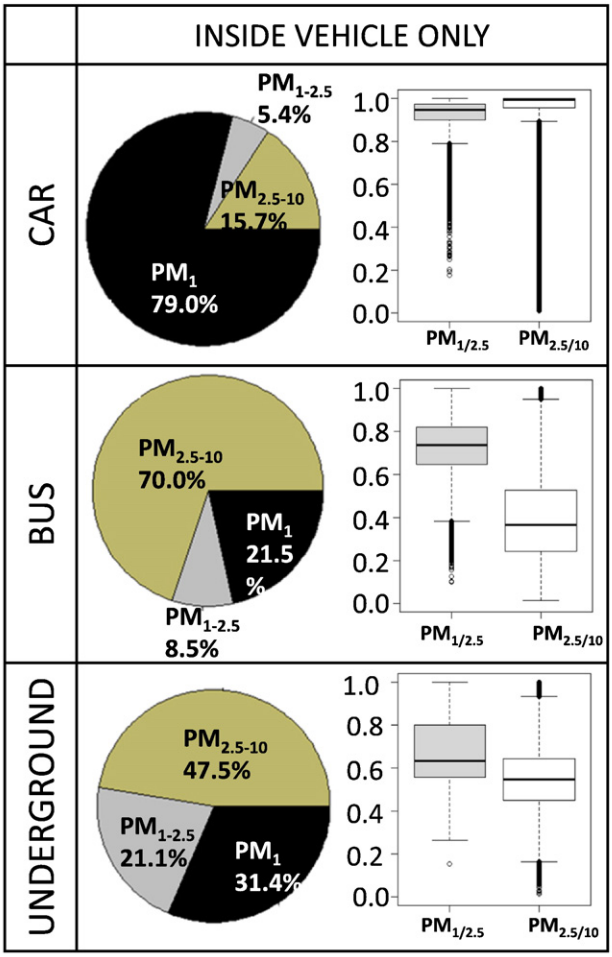

Fig. 3. Contribution of PM fractions to total PM, and boxplots showing the ratios $PM_1/PM_{2.5}$ and $PM_{2.5}/PM_{10}$ for each mode of transport (only the time spent inside the vehicles).

emissions per passenger, while bus passengers are disproportionally exposed to those emissions. This can be considered a violation of the core principle of environmental justice as reviewed by Brulle and Pellow (2006). Likewise, Jephcote and Chen (2012) reported that commuting of affluent population by private transport produced the high levels of air pollution in more deprived areas. Future studies are needed to evaluate exposure and deprivation at individual level for all the London population, in order to further explore the inequalities in exposure to air pollutants during commuting. The results from this study can be incorporated into predictive models of exposure during commuting.

3.1.3. Among period of the day

The highest concentrations of all pollutants were found during the morning trips, followed by evening and afternoon trips (Table 2). Afternoon and evening concentrations were not showing significant differences. Morning concentrations were 39% and 21% higher for BC, 19% and 13% for PNC, and between 35 and 43% and 23–30% for the different fractions of PM, in comparison with the afternoon and evening, respectively (Table 2). BC, PNC and PM_1, usually showed a peak during the morning hours, reflecting the influence of traffic emissions during this period (Costabile et al., 2009; Morawska et al., 2008; Moreno et al., 2009; Pérez et al., 2010). De Nazelle et al. (2012) also found significantly higher morning concentrations in personal measurements of commuters in Barcelona for BC and $PM_{2.5}$, although not for UFP. Similar results were observed by Gómez-Perales et al. (2007) in Mexico City for $PM_{2.5}$. On the other hand, Adams et al. (2001) and Kaur et al. (2007) reported no significant or little differences among the intra-day periods that they were studying, which was attributed to their routes covereing a small area close to the centre of the city.

3.1.4. Spatial variation

Fig. 5 shows the spatial variation of BC concentrations for selected evening car and bus routes. We were not able to generate the underground maps owing to the absence of GPS signal. Higher concentrations of air pollutants were expected in the central area of London, as shown for modelled $PM_{2.5}$ in the annual pollution map for 2010 (KCL's Environmental Research Group, 2016). However, the annual pollution maps also show the main roads and motorways in the city as hotspots of air pollution. The absence of an increasing gradient towards the city centre (Fig. 5) suggests that concentrations during the trips are mainly affected by the traffic conditions at each specific road. Previous studies have identified how traffic flow interruptions (such as red lights and other traffic intersections) result in higher PM mass concentrations and PNC in these points (Goel and Kumar, 2015b; Kim et al., 2013; Kumar and Goel, 2016). Therefore, concentrations inside the on-road vehicles were affected by the traffic flow and driving conditions of the specific point of the route.

The trips were performed on different days, so a direct comparison between income groups might be affected by the specific meteorology and traffic intensity of that day. However, we can consider Fig. 5 as an example of the overall results shown in Fig. 2, where this relationship among income groups for car and bus is maintained. For the bus trips, G_3 and G_4 (the routes starting from the two least deprived areas) clearly faced overall higher concentrations whereas G_1 and G_2 showed the lowest, especially G_2 (Fig. 5). This was probably because the longest section of G_3 and G_4 bus routes passed through central busy streets, were more traffic interruptions and higher background concentrations are present. A different pattern was observed for the car measurements, where the four income groups were exposed intermittently to high and low BC concentrations at each point of the route, but with G_2 showing a higher frequency of lower concentrations (green dots on the map, Fig. 5).

3.2. Identification of parameters governing the variability of concentrations in each transport mode

3.2.1. Bus measurements

Fig. 6 shows the overall concentrations of the different pollutants observed at different stages of the bus trips: (i) walking on the street, (ii) waiting at the bus stop or bus station or (iii) being inside the bus cabin. A similar pattern was followed by BC, PNC and PM_{10} with concentrations being the highest inside the bus and the lowest while walking on the street. This gradient is less marked for PM_1 and $PM_{2.5}$ concentrations, which were similar at the three locations (see Fig. S7 for PM_1 boxplot). This might be explained by the fact that PM sources are less specific for traffic emissions and are influenced by other transboundary sources (Cyrys et al., 2003; Kaur and Nieuwenhuijsen, 2009). However, resuspension of coarse particles (which affects PM_{10} concentrations) can be important at short distances as observed in previous work (Kumar and Goel, 2016). For the particular case of G_4, median PM concentrations of the three fractions were higher for the times when the field worker was waiting at the bus stop, probably indicating the influence of road dust resuspension.

3.2.2. Car measurements

Three different trip stages were assessed for the car measurements: (i) walking to the car, (ii) the period inside the car but with the engine still off (while getting everything ready), and (iii) the car trip itself. Similarly to bus measurements, BC and PNC showed highest concentrations when the fieldworker was inside the car with the engine on (this is, on the move; Fig. 6). On the other hand, and according to what has been discussed before, $PM_{2.5}$ and PM_{10} concentrations were reduced during the time spent in-cabin, especially for PM_{10}. PM_1 concentrations (Fig. S7) remained very similar across the different locations, which

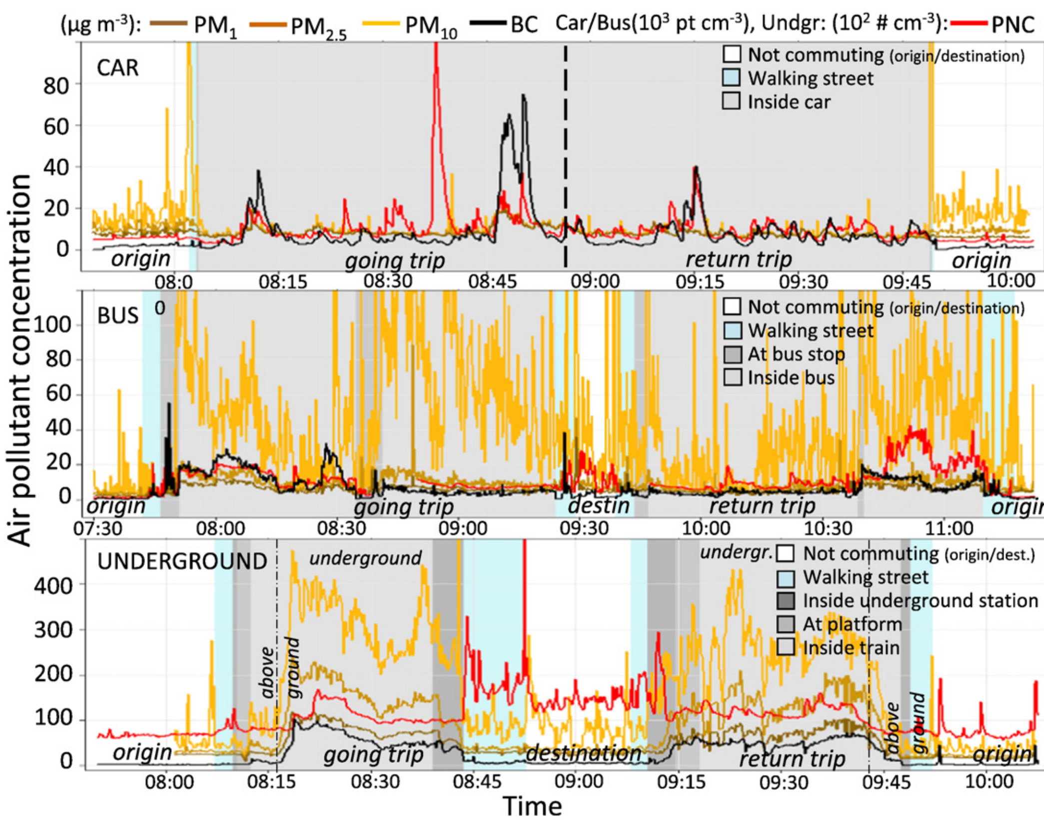

Fig. 4. Time-series for the car, bus and underground measurements corresponding to round (going + return) morning trip of the routes from the income group G_3. No measurements at the destination were made when commuting by car due to security reasons. Note the different units for PNC in the underground (10^2 cm^{-3}, rather than 10^3 cm^{-3}) and the different axis scale for each transport mode. BC measurements in the underground are overestimated because of Fe interference. Background colours indicate the location of the field worker.

may indicate a lower removal efficiency of the car filter for this fraction. The measurements in the car with the engine turned off correspond to much fewer data in comparison with the rest of locations since only a few minutes passed between getting into the car and starting the trip. This explains the very similar concentrations when being at the street since the in-cabin concentrations were directly affected by the opening of the doors at the moment of getting in.

3.2.3. Underground measurements

Four different stages in the underground trips were included in the analysis: (i) walking to the underground station, (ii) walking from the entrance of the underground station to the platform, (iii) waiting on the platform, and (iv) inside the train. Concentrations of PM became much higher from the moment that the commuter gets inside the underground station (Fig. 6; see Fig. S7 for PM_1). This situation was reversed for PNC, owing to the absence of combustion sources in this environment. The boxplots show a high dispersion of PM data in the underground environment owing to the numerous factors that affect concentrations. The impact of the underground lines has been already discussed in section 3.1 but other parameters also deserve attention, such as the influence of the above/underground factor in trains and platforms, and the effect of openable/non-openable windows in trains.

In the London underground, platforms can be underground or open and located at ground level (referred hereafter as above-ground). Underground platforms have much higher concentrations than the platforms above ground for all the pollutants (Table S8), owing to a hindered dispersion in the more confined underground environment. Particularly, the PM fractions were between 6.1- and 8.5-times higher in platforms underground. Much lower is the ratio for PNC, having 1.9-times higher concentrations in the underground platforms. For BC, this ratio increases to 30.0, but this is because the underground BC concentrations are overestimated due to Fe interference in the measurements while they are not on the platforms above-ground. Cartenì et al. (2015) found similar results in a study carried out in Naples. They obtained $PM_{2.5}$ concentrations between 45 and 58 µg m^{-3} (172–262 µg m^{-3} for PM_{10}) in stations above-ground in comparison to 10 µg m^{-3} (16 µg m^{-3} for PM_{10}) in a station at the ground level.

Air quality inside the underground trains is affected by the ventilation setting in the train (Cartenì et al., 2015; Martins et al., 2016a) and if the train is passing on an above or underground section (Aarnio et al., 2005; Adams et al., 2001; Nieuwenhuijsen et al., 2007). The trips on the District and Northern lines had sections on both above and under the ground while the part that we covered by Victoria line was always underground. The effect of the tunnels is obvious (Fig. 7; see Fig. S7 for PM_1), with much higher concentrations encountered in sections that go underground (GM: PNC = 7198 cm^{-3}, $PM_{2.5}$ = 63.0 µg m^{-3} for District line; PNC = 8524 cm^{-3}, $PM_{2.5}$ = 83.4 µg m^{-3} for Northern line) than above the ground (GM: PNC = 4204 cm^{-3}, $PM_{2.5}$ = 11.7 µg m^{-3} for District line; PNC = 3663 cm^{-3}, $PM_{2.5}$ = 6.61 µg m^{-3} for Northern line).

All the trains on the Northern and Victoria line had openable windows, which were generally all open. Trains on the District line have both openable (monitored on 21 single trips) or non-openable windows (26 single trips; the latter always had mechanical ventilation in operation). The concentrations in the trains with non-openable windows from the District line clearly show the lowest concentration (GM: PNC = 4549 cm^{-3} and $PM_{2.5}$ = 16.4 µg m^{-3}) in comparison with trains with openable windows (GM: PNC = 6624 cm^{-3} and $PM_{2.5}$ = 37.4 µg m^{-3}) for all the pollutants under study (Fig. 8). This indicates an easy entrance from the outside train particles into the carriage

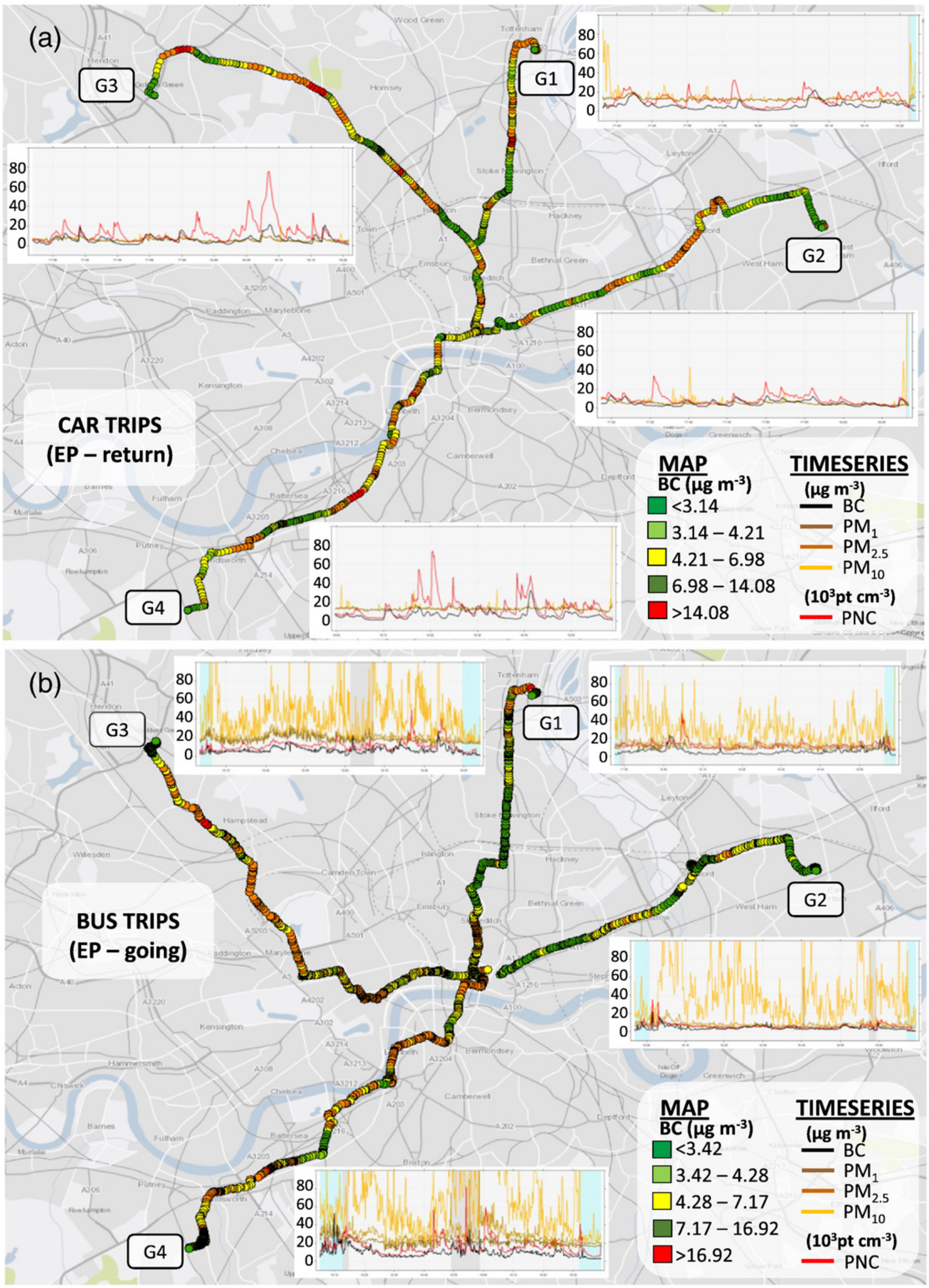

Fig. 5. Spatial distribution of BC concentration for (a) car and (b) bus during evening trips (each trip was carried out on a different day). The time-series for the corresponding trip are shown.

through the windows, probably enhanced by the turbulences created by the train itself. Since the concentrations were higher in the tunnel sections, the synergy of being underground and having the windows open leads to the very high concentration found in the Northern and Victoria lines. These results are in line with other studies assessing the ventilation condition. Martins et al. (2016a) carried out a comparison among the underground systems in three European cities and observed increased $PM_{2.5}$ concentrations in the trains from Athens, which had the windows open. A gradual substitution of the trains with openable windows by non-openable windows trains would be advised to substantially diminish the exposure of the underground passengers.

Still, differences can be observed among the trains on different lines with openable windows (Fig. 8), with the Victoria line being the most polluted for the fine fraction of particles ($PM_{2.5}$ and PM_1; see Fig. S7

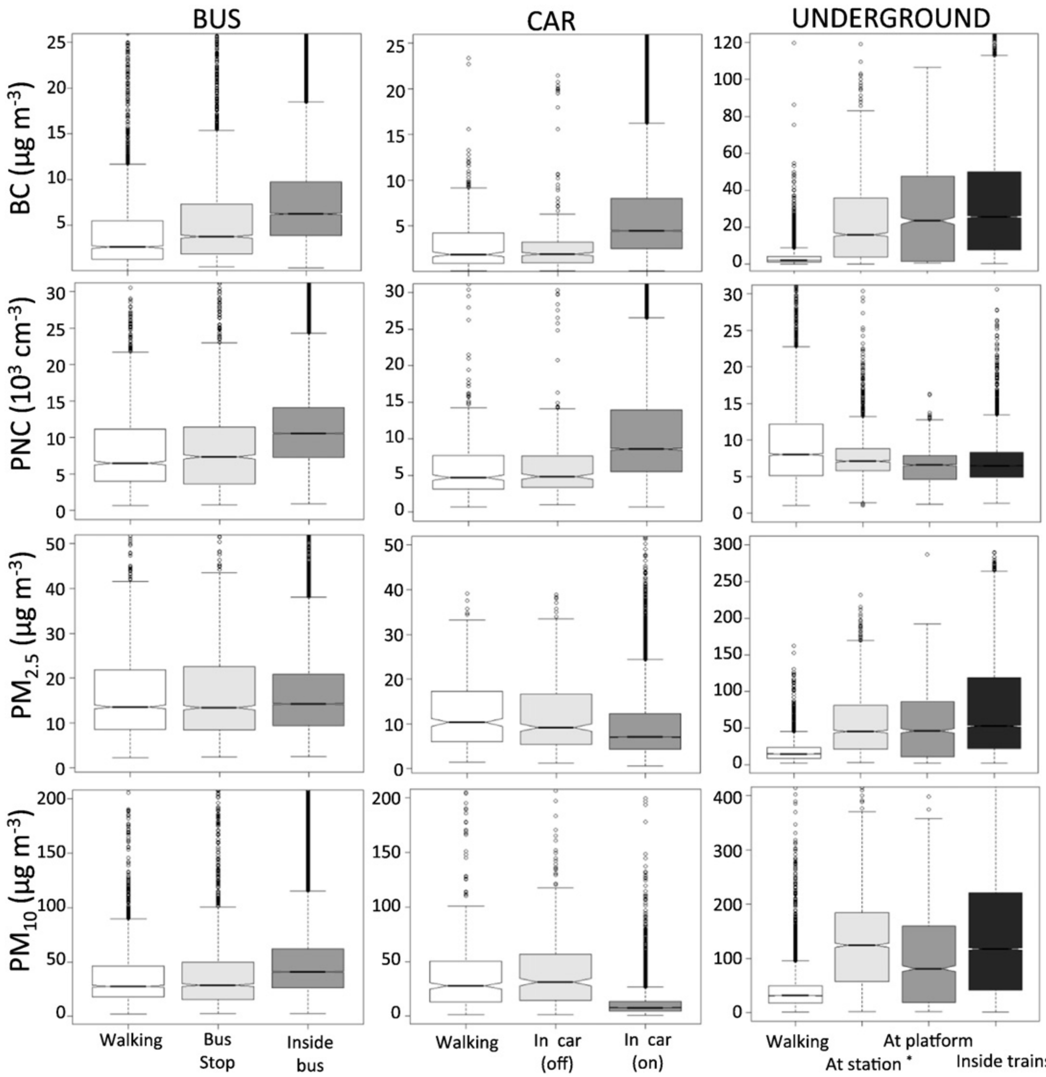

Fig. 6. Boxplot of BC, PNC, $PM_{2.5}$ and PM_{10} concentrations measured at the different location during the trips by each income group and different transport modes. Boxes represent 25th and 75th percentile, central line the median, bars outside the box represent the 1.5× interquartile range, and circles are outliers. The notch displays the confidence interval around the median. *Time spent in the underground station but excluding measurements at the platform. *In-car* (*off*) = inside the car with the engine turned off; *In-car* (*on*) = inside the car with the engine on (i.e. on the trip).

for PM_1). However, for PNC and PM_{10}, the Northern line is showing the highest concentrations. A possible explanation would be the length of the underground section that might lead to a higher accumulation and subsequent resuspension of the particles. However, this does not agree with our results. In our measured section, Victoria line goes above ground more often than the Northern, so the highest PM would be expected (all fractions, not only PM_{10}) in Northern (owing to the lower dispersion of this underground-generated particles).

3.3. Duration of the trips

Underground was the fastest mode (42.6–55.9 min; Table S7), with 40% of the time being spent in reaching the underground station and waiting for the train. The second fastest mode was car (49.4–66.3 min) since we selected the most direct routes. Although the fastest options were selected, bus trips took by far the longest time (67.4–108.1 min) due to the indirect and busy central routes and the frequent stops. Reaching the bus took 18% of the total time, which in absolute terms is similar to the underground (15.9 min and 18.6 min for bus and underground, respectively; Table S7). Since the distance between origin and destination was longer for less deprived areas (Table 1), G_1 trip duration was always the shortest among all modes. G_4 trip duration was the longest for car and underground, while it was G_3 for the bus.

3.4. Contribution of each sub-microenvironment of the trip to the overall RDDs of $PM_{2.5}$

The concept of exposure incorporates the duration of the contact to a certain concentration by integrating over time (Duan, 1982; Ott, 1982). The estimation of doses go a step further and includes a dosimetry factor such as the ventilation rate (Morawska et al., 2013) and can also incorporate the fraction of particles that are deposited in the lungs. We estimated the RDD per hour of exposure at each trip stage and the total RDD per trip for females (31–40 years old), for each transport mode and income group (Table S7). The different RDD observed between females and males is due to the different ventilation rates assigned to each of them (Table S3; US-EPA, 2009), being between 23 and 32% higher in males. For the sake of brevity, we will only refer to the females RDD in

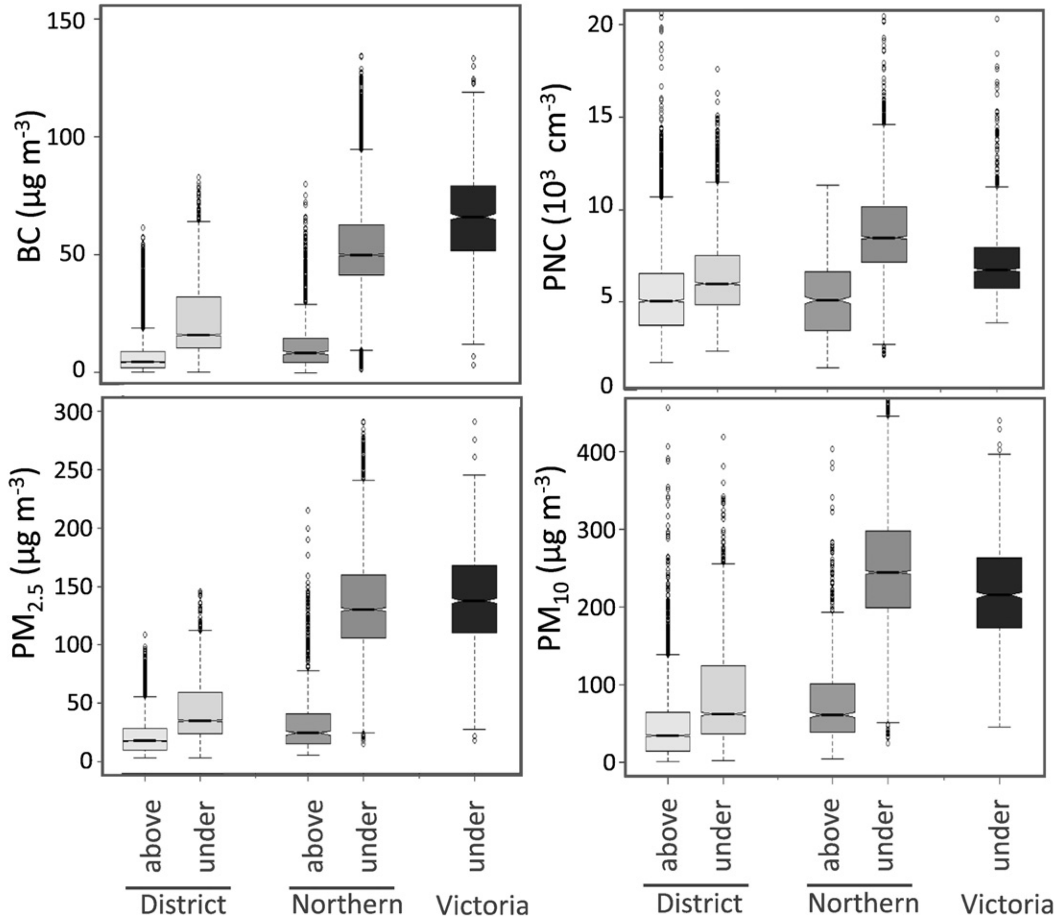

Fig. 7. Boxplot of BC, PNC, $PM_{2.5}$ and PM_{10} concentration measured at the underground in different lines, separated by above or underground section. Boxes represent 25th and 75th percentile, central line the median, bars outside the box represent the 1.5× interquartile range, and circles are outliers. The notch displays the confidence interval around the median.

this work but a very similar pattern is followed by the male results (Table S7). Different RDD are obtained for each PM fraction. A discussion about these differences is included in the Section S1 in the Supplementary Information.

The contribution of each stage of the trip to the total trip RDD varies substantially among transport modes (Fig. 9). The hourly RDD while walking varied among the groups and transport mode (the walking itinerary was different for each transport mode) ranging from 2.79 to 4.76 μg h^{-1} (Table S7). The in-vehicle RRD for car was the lowest (0.57–0.95 μg h^{-1}), followed by RDD in the bus (1.52–3.49 μg h^{-1}). The highest hourly RDD was found in the underground, both inside the station and in the trains. Walking inside the stations gives RDD from 9.34 to 21.03 μg h^{-1}, which is a bit lower while waiting at the platforms (5.95–18.17 μg h^{-1}) since lower ventilation rates due to lower activity intensity. The in-vehicle hourly RDD for the underground was 4.10 and 4.50 μg h^{-1} for the District line (G_4 and G_2 respectively) and >5-times higher in the Northern line (21.63 μg h^{-1} for G_1 – includes Victoria line; 25.8 μg h^{-1} for G_3).

The doses received (and the time spent) during the in-vehicle for the car routes were between 92.6 and 96.1% (97.4–99.3%) of the total dose (time) received during the whole trip, with the remaining 4.0–7.4% (0.7–2.6%) corresponding to walking on the street. The walking distance from the origin point to the car was always short, and therefore also was the time and the dose. The contribution of the in-vehicle to the total trip doses for the bus routes was also relatively high (68.0–78.5%; time: 78.4–87.1%), with the walking contribution (15.3–19.8%; time: 7.1–13.7%) gaining some importance because of longer times needed to reach the bus stop. Waiting at the bus stop/station contributed with the 5.9–12.2% of the total trip RDD (time: 5.6–12.1%). On the other hand, the contribution to the total RDD per trip from the time spent in the underground trains was generally lower (53.8–66.0%, except for G_3 which was 80.1%) since the time spent inside the vehicles for this mode was also relatively short (49.8–66.9% of total time). The contribution to the total trip dose of walking on the street showed a high variability (4.4–21%; time: 18.0–27.8%) owing to the different distance from the origin or the destination to the underground station and, particularly, to the high relative contribution of the in-vehicle time for Northern line routes (G_1 and G_3). The time spent inside the stations (including walking inside the station and waiting in the platform itself) implies receiving between the 15.6 to 30.8% of the total trip RDD (time: 10.6–27.2%). Due to the very high concentrations in this environment (Table 2), the absolute contribution (in terms of μg) of the in-vehicle for the underground is often larger the total trip RDD for the rest of the transport modes. The contribution of out of vehicle locations in the public transport modes (bus and underground) was noteworthy. The time spent and doses received could be reduced by increasing the buses and trains frequencies, especially in the peak commuting hours which is when the highest background concentrations are observed.

4. Summary, conclusions and future work

We investigated inequalities related to income deprivation in the exposure to PM_1, $PM_{2.5}$, PM_{10}, BC and PNC during commuting, as well as assessed the differences between transport modes (car, underground and bus) and daytime periods. For the first time, inequalities were assessed through direct exposure assessment. We selected different routes, linking a place of residence with distinct levels of income

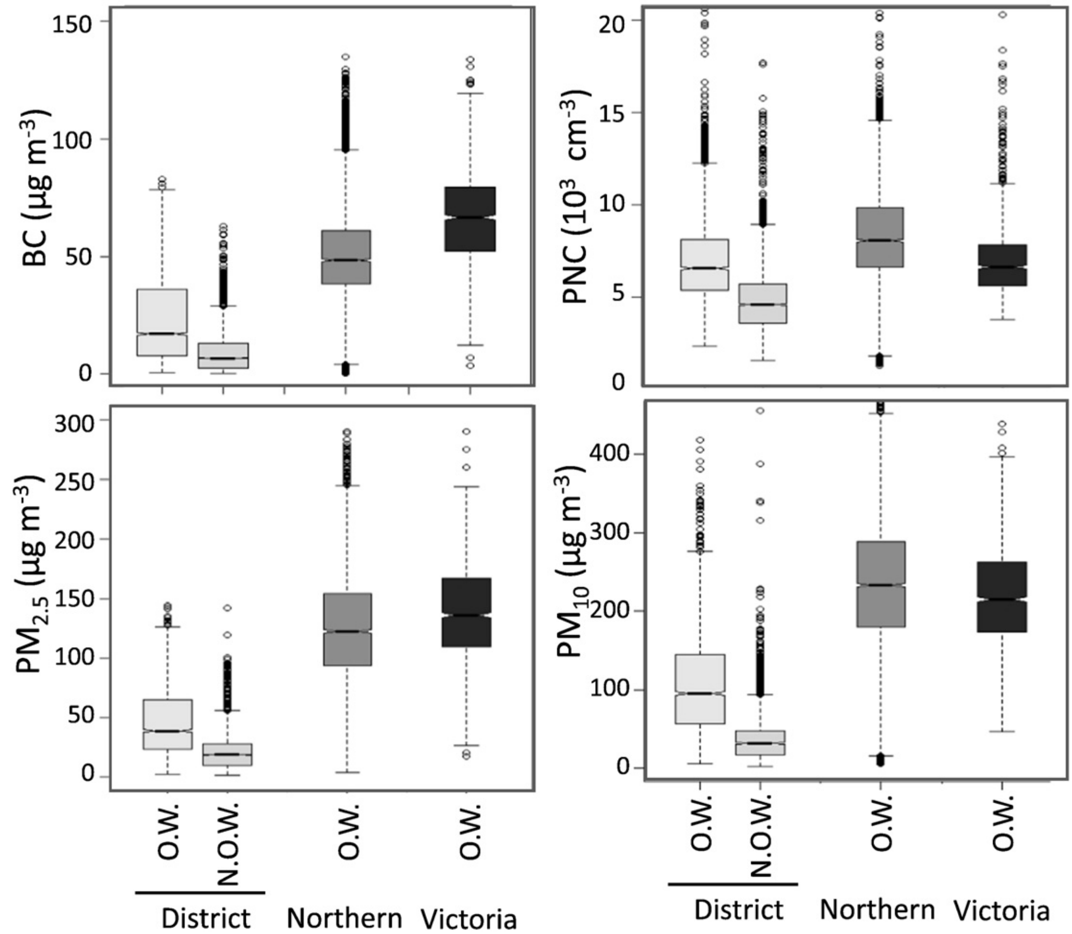

Fig. 8. Boxplot of BC, PNC, $PM_{2.5}$ and PM_{10} concentration measured at the underground in different lines and in trains with different ventilation settings. O.W. = Openable windows, N.O.W. = Non-openable windows (mechanical ventilation). Boxes represent 25th and 75th percentile, central line the median, bars outside the box represent the 1.5× interquartile range, and circles are outliers. The notch displays the confidence interval around the median.

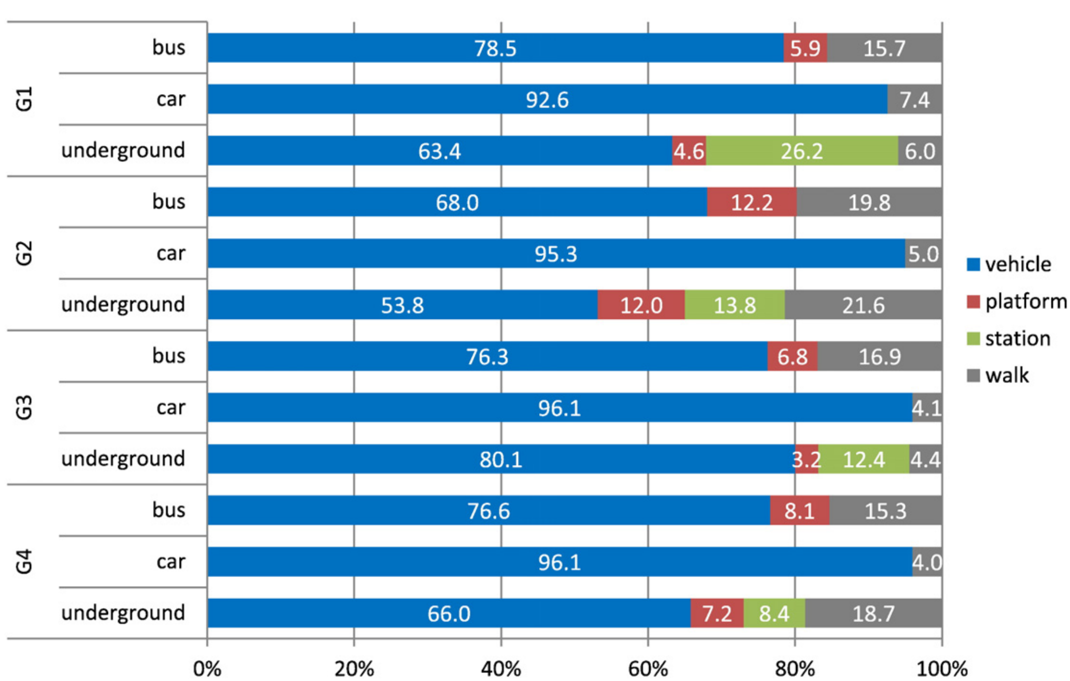

Fig. 9. The relative contribution of each of different part of the routes to the total RDD per trip corresponding to $PM_{2.5}$.

deprivation (G_1 to G_4, from most to least deprived) to a place of work (financial district in London). We conclude that:

- The most deprived income group, G_1, showed the overall highest concentrations of BC and all PM fractions while G_3 showed the highest PNC. G_2 had the lowest concentrations of all pollutants. There was no consistent decrease or increase in concentrations related to the income deprivation of the origin.
- The highest concentration of all PM fractions was found in the underground measurements, followed by bus and car. BC concentration was higher in the bus (5.4 $\mu g\ m^{-3}$) than in the car (4.4 $\mu g\ m^{-3}$). Car filtering system was efficient in hindering the entrance of PM, especially coarse particles.
- PNCs were highest in the bus (9355 cm^{-3}), not statistically different from the car (8639 cm^{-3}). The absence of combustion sources made the underground trips to have the lowest PNCs (6694 cm^{-3}).
- In general, the concentrations of all the pollutants in the road modes (car and bus) were ruled by traffic conditions and flow interruptions at the specific road section.
- Substantially higher concentrations of all pollutants, especially PM, were observed in underground carriages with openable windows and in the underground sections. An upgrade to trains with non-openable windows would be advised to substantially reduce passenger's exposure.
- Concentrations of pollutants during the morning peak were between 13 and 43% higher than during the afternoon or evening peak, depending on the pollutant.
- There is a violation of the core principle of environmental justice (Brulle and Pellow, 2006) if we consider that commuters from less deprived areas rely more on the car (low exposure) and those from more deprived areas on the bus (high exposure). Besides, private emissions generate the highest emissions per passenger.
- Bus trips took the longest time (67–108 min) due to indirect routes and frequent stops. Underground was the fastest mode (43–56 min), followed by car (50–66 min).
- For the public transport modes, there was a considerable out of vehicle contribution to the total RDD. The time spent walking and waiting inside the underground stations contributed to 15.6–30.8% of the total RDD while waiting time at the bus stop contributed to 5.9–12.2% of total RDD. Exposure might be reduced by increasing the trains and buses frequency, especially during peak hours.

Whilst concentrations have been determined for typical routes for 4 income groups and for the most prevalent transport modes, further experimental campaigns and extrapolation studies are needed to determine possible inequalities on the exposure to air pollutants in the wider urban system.

Acknowledgements

This work was supported by the Economic and Social Research Council [grant number ES/N011481/1]. ASTRID (Accessibility, Social justice and Transport emission Impacts of transit-oriented Development) project was jointly funded by the ESRC (UK), NWO (Netherlands) and FAPESP (Brazil). We thank Louise Dearman and Olivia Carlton for their kind support of our monitoring work in London underground.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2017.01.019.

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